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Management of Remnant Prairie Ecosystems: Soil Transfer, Fire, and Exotic Species Invasion

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MANAGEMENT OF REMNANT PRAIRIE ECOSYSTEMS:

SOIL TRANSFER, FIRE, AND EXOTIC SPECIES INVASION

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ABSTRACT

Warrix, Adam R. M.S., Purdue University, May 2016. Management of Remnant Prairie Ecosystems: Soil Transfer, Fire, and Exotic Species Invasion. Major Professor: Jordan M. Marshall.

Property management is an important aspect in sustaining the Earth's ecosystems. Anthropogenic changes to the planet have made conservation important in maintaining species abundance and diversity as well as limit damages to sensitive ecosystems. Invasive species have become a standard, consistent problem in managed ecosystems. Such species often invade into disturbed areas, which commonly are a result of restoration or other management activities. In dredging open water bodies at Eagle Marsh Nature Preserve (Allen County, IN), artificial mounds were constructed with the soil removed. This movement of soil created a disturbance which facilitated for the recruitment and colonization of invasive species. The purpose of this study was to quantify the effectiveness of transferring a native seed bank in a mesic prairie to the artificial mounds as a method of controlling invasive species. The native seed bank in the mesic prairie would be lost due to additional management. This study was conducted with greenhouse and field experimentation. Soil samples containing the seed bank were moved to a greenhouse to quantify plant emergence following different intact and mixed soil treatments. All emerging plant species were identified and counted. All treatment

methods contained a high diversity of native species and a low diversity of invasive species with the exception of the seed bank soil mixed with soil from the mounds. The mound mixed soil contained a higher diversity of invasive species than other treatments. A plant survey of the mounds was conducted in July 2014. In October 2014, 960 m^2 of soil was transferred from the donor site to the mounds. A post-move survey conducted in July 2015 indicated the native species diversity increased while the diversity of invasive species decreased. Soil transfer as a management strategy was appropriate in this situation where a large native seed bank in a donor region existed. By moving soil from one location to another, we were able to promote native plant establishment and reduce invasive species emergence. While soil transfer is effective, other management strategies may not have the same results. Callery pear (*Pyrus calleryana*) is an invasive tree introduced to North America as an ornamental from southwest Asia. It easily invades disturbed areas, causing a disruption to mid- to late-successional species establishment. The purpose of this study was to assess Callery pear demographics in a managed prairie and quantify the effects of a prescribed fire management strategy on Callery pear density and recruitment. This study was conducted at Arrowhead Prairie (Allen County, IN), managed by Little River Wetlands Project (LRWP). Before 2009, Arrowhead Prairie was primarily used for agriculture. Following LRWP acquisition, the property has undergone active management including native plant seeding and prescribed fire. The prairie was divided into a north and south section using a historic ditch, with the south section burned in April 2014 and the north section burned in May 2015. Bare seeds, fruit, first-year seedlings, and second-year seedlings were also burned in a separate controlled experiment. Fire in both years top killed Callery pear individuals, with 100% of surveyed

trees producing epicormic shoots and 83% producing more than one epicormic sprout. First-year seedlings exposed to the fire had a 3% resprout rate while all seconds-year seedlings resprouted. In contrast, 13% of individuals at Arrowhead Prairie had epicormic shoots without fire. Trees were significantly shorter post-burn than prior to the burn; however, root collar diameter was not different, suggesting that resources were used in primary growth. While fire did reduce stem height, it did increase the number of stems occurring within the burned section due to epicormic sprouting. Root stored reserves provided necessary energy to produce numerous sprouts by the majority of pear trees, although no trees produced flowers during the duration of the study. Plant age likely plays an important role in response to fire, with seeds and fruits succumbing. However, older seedlings may survive with increased epicormic shoot production. Multiple techniques combined with relevant studies allows for land managers to make appropriate decisions in terms of management. Although many of these techniques are not universal, a wide range of researched techniques provides multiple avenues of approach when managing a parcel of land.

INTRODUCTION

Property management is an important aspect of the continuation of the Earth's ecosystems. Proper land management can enable continued sustainability while preserving aesthetics of land and is a continuous process (Aide et al. 2001, Hobbs and Harris 2001). Removal of invasive species, plantings of native species, cleaning of litter and debris, and proper application of disturbance must be repeatedly applied (Hobbs and Norton 1996). Many of these may not apply to many ecosystems, but by properly identifying key characteristics for each ecosystem, land managers can make more informed decisions. Although attempts have been made to apply management at the landscape level, much of management is done on a localized, small-scale (Palmer et al. 1997). This is due to land owners not owning a landscape level parcel and the cost of such restoration (Palmer et al. 1997). Because of these limitations, small scale application of management is important.

Much of the information about long term restoration is information collected from various projects and pooled together into reports (Wu 2006). Looking at information from a multiple ecosystem environment can also increase the data in which these reports draw from (Palmer et al 1997). A problem exists in combining this information to make large-scale reports. The pool of information can be so great that needed information is lost (Palmer et al 1997, Wu 2006). By reducing the amount of information to a manageable level, land managers can take actions appropriate to their parcel.

The wealth of information gives land managers multiple management techniques. Although many are available, some of these techniques are very ecosystem specific. Soil transfer is a common management strategy for wetland and prairie restoration (Brown and Bedford 1997, Bruelheide and Flintrop 2000). Soil transfer has many benefits which can be exploited in both ecosystems (Brown and Bedford 1997, Bruelheide and Flintrip 2000). Many of these benefits may also be available to other ecosystems but without valuable research, this knowledge would not be available (Hobbs and Harris 1996, Wu 2006). Simply preserving the aesthetic value of an ecosystem is important for some land managers, which may limit the ecosystem services provided by insignificant amounts (Hobbs and Harris 1996, Boyd and Banzhad 2007). By engineering an ecosystem to provide both aesthetics and services, two objectives of management can be accomplished (Boyd and Banzhad 2007).

Because of the potential ecological and economic impacts, many management techniques involve controlling non-native species. Although the exact definition of invasive species is debated, a common definition seen is that an invasive species is any species living outside it native range and cause ecosystem disruption often by displacing native species (Colautti and MacIssac 2004, Gurevitch and Padilla 2004, Clout and Williams 2009). It has been hypothesized that invasive species can cause extinction but much of the evidence is anecdotal and holds little merit (Ricciardi 2004). The impact which invasive species cause may not be positive. However, in some ecosystem, invasive species have cause an increase in overall diversity or may be neutral in impact

(Rosenzweig 2001, Marshall and Buckley 2009, Warrix et al. 2015). Although the effects of invasive species can vary, their disruption to ecosystem functions are problematic and are worthy of study.

Invasive species follow patterns, allowing for prediction of spread but provides little insight into management (Kolar and Lodge 2001, Dwyer and Morris 2006, Drury et al. 2007). While prevention is the best policy, an established invader needs proper management for removal (Williamson and Fitter 1996, Zavaleta et al. 2001). Management of invasive species cover a wide range from simply removing the species by hand to an ecosystem wide removal through chemicals or controlled burns (Zavaleta et al. 2001, Clout and Williams 2009). Many of the techniques are species specific due to the species having a specific morphology as well as a limited amount of knowledge of control (Clout and Williams 2009). For a species like Callery pear (*Pyrus calleryana* Decne. [Rosaceae]), much of the knowledge of management comes from planted cultivars and the various conditions in which they can survive (Culley and Hardiman 2007).

One strategy to change species composition and promote specific species is soil transfer: the process of moving soil from one location to another with the goal of maintaining the soil contents (Brown 1998). Soil contents include seeds, established plants, organic material, fungi, ground- and soil-dwelling animals, microorganisms, and other matter existing within the soils (Brown and Bedford 1997, Brown 1998). Soil type is maintained during the transfer allowing a change of the soil composition at a recipient location (Brown and Bedford 1997). Soil can also be transferred by multiple means. Sod cuts are used to keep the soil as intact as possible while moving whereas a front-end loader and a dump truck will mix the soil during the transportation process (Brown and Bedford 1997, Weber 1992). The process in which the soil is moved is highly dependent on the objective of the transfer and plant community present (i.e. sod cuts require dominance of grasses).

Fire is a common ecological disturbance in many ecosystems and is a widely used management technique (Neary, et al., 1999). Like soil transfer, fire is useful in promoting specific groups of species. Irregularity of fire contributes to the effect that it has on an ecosystem, varying in frequency, season, and intensity (DeBano, et al., 1998). These three factors combine to determine the fire regimes and can be used to create management strategies in different ecosystems (Whitlock, et al., 2010). One strategy is to use fire to control invasive species while promoting native plants which may require fire to break dormancy of seeds (DeBano et al. 1998, Keeley 2006). Even though fire has been well studied, this is still an area in which knowledge gaps exist.

Lightning is the most common cause of grassland fires (Renkin & Despain, 1992). Many grasses produce seeds that rely on fire, which serves to release them from dormancy and to prepare the soil for the seeds by altering resources and reducing competition (Ojima, et al., 1994). Prairies can be colonized by forest species, but fire typically keeps these areas open and disturbed to allow for herbaceous, prairie species to dominate (Brooks, et al., 2004).

Fire suppression occurs when a land manager removes the build-up of flammable debris or suppresses an active fire (Aplet 2006). As a form of disturbance, fire suppression can cause a reduction in diversity of species which need fire to survive, both plant and animal (Renkin and Despain 1992). Historically, fire has been seen an having a negative result on landscapes, but recently the benefits of fire has been noticed allowing for many agencies to implement policies promoting controlled burns (Wagtendonk 2007).

Callery pear (*Pyrus calleryana* Decne. [Rosaceae]) is an introduced tree in North America originally from Asia and widely planted as an ornamental. It was originally introduced due to its resistance to fire blight, a bacterial disease, and became a common ornamental tree in the 1950's, which is when many herbarium samples began appearing (Reimer 1925, Creech 1973, Culley and Hardiman 2007). It has spread throughout much of the eastern United States and establishes in disturbed areas (Vincent 2005, Culley and Hardiman 2007). Vincent (2005) found that Callery pear exhibits improved establishment and recruitment in mid- to late-successional habitats. However, as distance from roads increases into forests, Callery pear density decreases (Flory and Clay 2006).

Callery pear is a deciduous tree, which can grow to 20 m in height. The leaves are alternate, simple, broad-ovate to ovate; 4 to 7 cm long and waxy, dark green in color. In the autumn, leaves turn dark red in color and stay attached to the tree for an extended period. The flowers are white and contain five petals with a 2-3 cm diameter corolla (Reimer 1925). Trees can reach maturity in three years, producing flowers in the early spring which exhibit a strong smell (Culley and Hardiman 2007). The fruit is a small, hard, round pome less than 1 cm in diameter, and the woody fruit exterior is usually softened by frost (Vincent 2005). Some cultivars were thought to be sterile due to self-incompatibility of this species (Culley and Hardiman 2007).

There have been few studies on the ecological effects of Callery pear, although it has been predicted that they can impede establishment of mid- to late-stage successional species (Culley and Hardiman 2007). Thorny thickets can form causing a management problem, if the thorny wild-type is present. Callery pear can be girdled early in the growing season. Mowing of smaller trees has shown to be ineffective as the trees easily produce epicormic sprouts (Culley and Haridman 2007). Fulcher (2002) noted that many Callery pear trees planted in urban environments are susceptible to limb breakage. This, combined with the litter of the fruit on the ground, can cause both an unpleasing aesthetic effect and a potential danger to foot traffic (Dirr 1998, Fulcher 2002). Even with such few studies of impacts, Callery pear has been listed by Indiana DNR as invasive in natural areas and as a “high” rank by the Nature Conservancy and Indiana Invasive Species Council (Homoya 2010, Jacquart 2012, IISC 2013).

The overall objective of my thesis was to test the effectiveness of two management strategies (soil transfer and prescribed fire) on controlling invasive species within specific management scenarios. Chapter 1 of my thesis contains the results of an investigation into the effectiveness of using soil transfer as a management strategy to control invasive species and promoting native species. In the case of Chapter 1, an established native mesic prairie seed bank was moved to a site that had been colonized by invasive plants, which were persisting despite other management attempts. Chapter 2 focuses on the response of established Callery pear individuals to prescribed fire and identifying the effectiveness of fire as a management strategy. This invasive tree had colonized a managed prairie fragment that was scheduled to be burned and the susceptibility of Callery pear to fire was unknown. Chapter 3 is an extension of the previous chapter and presents a controlled experiment regarding seed and seedling sprouting following fire. Since Chapter 2 was centered on older, established trees, I was interested in quantifying seed and young seedling survival after fire.

CHAPTER 1. USE OF SOIL TRANSFER WITH A NATIVE SEED BANK TO CONTROL INVASIVE PLANT SPECIES IN A MANAGED MESIC PRAIRIE ECOSYSTEM

Introduction

Soil transfer is a common method used to move a seed bank or plants from one place to another while preserving the contents of the soil (Elliott and Coleman 1988, Brown and Bedford 1997, Waldrop and Firestone 2006, Hölzel and Otte 2009). Movement of soil transfers more than just the seed bank and plants, a host of other microorganisms and fungi are also moved (Elliott and Coleman 1988, Waldrop and Firestone 2006). This can potentially reduce the risk of invasive species recruitment while promoting natives and more rapid establishment from the seed bank (McKnight 1992, Brown and Bedford 1997). Soil transfer is common for wetlands since the soil type and seed bank allows for a near seamless restoration (McKnight 1992). Because wetland soils develop under special conditions (i.e. saturated with water, reduced oxygen), transferring these soil types reduces the time for soil formation in restoration efforts (Mitsch and Gosselink 2000). With proper management after a soil transfer, a restored environment can be establish (Brown and Bedford 1997). In practice, this appears to be a viable management; however, seed bank content has been a poor predictor of plant species abundance (Brown 1998).

Soil transfer has also been used to move prairie species as sod cuts in grass dominated ecosystems (Weber 1982, Kearns 1984, Bruelheide and Flintrop 2000, Conlin and Ebersole 2001). Sod cuts move the soil with minimal disturbance from one site to another (Weber 1982). This is effective in that it preserves the structure of the soil, although it can be expensive due to the requirements for cutting the soil to ensure that it remains intact in addition to the movement cost for the soil (Weber 1982, Bruelheide and Flintrop 2000). In addition to cost, sod cut soil transfers are only possible in grass dominated communities where root systems hold soil and cut strips intact. The effectiveness of soil transfer has not been properly assessed in forb dominated prairie ecosystems, even as it becomes are more widely used management strategy (Weber 1982, Bruelheide and Flintrop 2000). Even with these studies, the movement of soil may play a role in which species are transferred in the seed bank (McKnight 1992, Brown 1998, Bruelheide and Flintrop 2000).

During the restoration of wetlands, mounds can be created from dredging (Pethick 2002, Stevens et al. 2002, Hinkle and Mitsch 2005). These mounds play an ecological role which are complimentary to the wetlands, such as providing refuge to amphibious animals and ground-nesting birds (Stevens et al. 2002). Few studies have focused on these small areas with increased elevation neighboring wetlands (Stevens et al. 2002). Proper restoration and management of these mounds is important in reducing the number of invasive species while promoting native species (Steven et al. 2002, Clout and Williams 2009). An effort should be made to quantify the effectiveness of soil transfer on these mounds, since there is a lack of studies in both areas. The objectives of this study were to 1) identify the species present in a donor seed bank soil, 2) quantify the impact of soil movement techniques on seed bank recruitment in a recipient location, and 3) test the hypothesis that soil transfer is an effective management strategy for colonizing disturbed areas with native plants.

Materials and Methods

Study Site

Eagle Marsh is a 190 hectare nature preserve in southwest Allen County, IN (UTM 16N 4544480 648655). Little River Wetlands Project (LRWP) acquired the property in 2005 and restored larger areas of the property to wetland. Prior to this, the property was drained extensively and used for row crop agriculture (personal communication, B. Yankowiak). Immediately after acquisition by LRWP, several open water bodies were constructed by USDA NRCS (personal communication, B. Yankowiak). As a by-product of dredging those water bodies, mounds were developed, which have been colonized by numerous invasive plant species. LRWP attempted to control the invasive species by hand sowing seeds on five occasions, drill seeding on three occasions, and continuous weed removal. These attempts appeared to be unsuccessful. Native mesic prairie seed stock was planted on the west side of the Graham McColloch Ditch to promote native plant diversity in an area of moist, but well drained soil (Figure 2.1).The native mesic prairie area was set to be destroyed as part of further management and development of the property by LRWP as part of a federally funded project to control Asian carp. The potential destruction of the seed bank was the impetus for this study to understand transferring seed banks from an established donor area to a recipient area.

Greenhouse Experiment

In order to quantify the species present in the native seed bank of the donor soil, a greenhouse study was performed to identify species present and allow for the multiple possibilities in which the soil may be transferred. A 20 meter spaced grid was generated over the donor area at Eagle Marsh using ArcGIS (ESRI, Redlands, CA, version 10.2.2) (Figure 2.1). From this grid, seven plots were randomly selected and soil was removed from each plot. Six samples of soil were taken from each of plot, one for each treatment applied, by cutting an approximately 25 cm x 52 cm rectangle in the soil surface with a sharpshooter shovel and scooping the sample with a flat grain shovel at approximately 15 cm deep on 17 April 2014. Treatments included an intact sample of soil with organic litter (Intact), an intact sample of soil which was completely inverted (Intact Inverted), an intact sample of soil with the organic litter removed by hand (w/o Organic), a sample of soil mixed homogenously by hand (Mix), a sample of soil mixed homogenously by hand with an equal amount of soil from the recipient site (Mound Mix), and similar intact, invert inverted, mix, and mound mix sample of soil with organic litter removed by controlled fire. The treatments were placed into greenhouse trays measuring 25.5 cm x 52 cm x 6.5 cm. Controlled burning of the site occurred on 18 April 2014 and fire treatment samples were collected from three randomly selected plot locations on 21 April 2014. After collection, trays were transported to a greenhouse maintained at 21 °C and approximately 33 percent relative humidity, with ambient light. Day length was extended to 18 hours with sodium vapor lamps.

All plants were identified to species and counted with nativity determined using the USDA PLANTS Database (USDA NRCS 2016). After 4 months and all species were identified, the soil was dried at 50 °C to a constant weight to quantify soil volume within each treatment and replicate. Species richness and diversity were calculated at the tray level. Diversity was calculated as Shannon entropy index (H' = $\Sigma p_i \ln p_i$, where p_i is the proportion of the *i*th species within the tray). An ANOVA was used to analyze the difference in diversity between treatments for native and introduced species, as well as for total diversity. Trays which were collected after the fire were separated to include fire as a treatment. A Sørensen index was used to identify similarities between treatments. To analyze differences in soil weights and stems/kg, an ANOVA was performed with a Dunnet's post-hoc test using the Intact treatment as the control group.

Field Experiment

A comparison of the mounds pre- and post-move was necessary to quantify changes in the plant community as a result of the soil transfer. A 5 meter spaced grid was placed over the two recipient mounds using ArcGIS (ESRI, Redlands, CA, version 10.2.2) (Figure 2.1). Eight plots per mound were randomly selected and I conducted a plant survey at each plot location within a 1 m^2 quadrat during July 2014 (Pre-Move). All plants within the quadrat were identified to species and counted with nativity being determined using the USDA PLANTS Database (USDA NRCS 2016). Soil was moved from the donor area to the recipient area on 9 October 2014. Prior to transfer, a bulldozer cleared a 30 m x 32 m area to approximately 15 cm deep on 29 September 2014. Soil was moved via front end loader and dump truck to the recipient area. During August 2015, I conducted a similar survey at eight newly selected points on each mound (Post-Move).

Species richness and diversity were calculated on the quadrat level. Diversity was calculated as described above. A Student's t-test was performed to measure a difference between native and introduced diversity before and after the soil transfer. A Sørensen index was used to assess the similarities between the two mounds both prior to and after the soil transfer as well as the similarity between the pre- and post-move mounds. A species accumulation curve was calculated using the vegan package in R (Oksanen et al. 2015). All statistical analyses were conducted in R using R Commander (Fox 2005, R Core Team 2015).

Results

Forty-three species were encountered across all greenhouse experiment treatments (Appendix A). Of these species, fourteen were seeded by LRWP in previous years. Twenty-seven species were native species, with the remaining sixteen identified as introduced. Fire did not cause a significant difference in diversity between native, introduced, or pooled species (F=1.04, d.f.=1,31, p=0.316; F=0.05, d.f.=1,31, p=0.828; F=0.23, d.f.=1,31, p=0.637; respectively). There was no significant difference between the native diversity across the trays (F=3.21, d.f.=4,31, p=0.227) (Figure 2.2). Introduced and pooled species diversity had a significant difference among treatments (F=3.21, d.f.=4,31, p=0.026; F=2.88, d.f.=4,31, p=0.039; respectively) (Figure 2.3 and 2.4, respectively). A Dunnett's post-hoc test resulted in all treatments similar to the Intact treatment for the introduced and pooled diversity ANOVA with the exception of the Mound Mix treatment. A Sørensen index indicated that all treatments had a 70% or greater similarity when compared to each other with the exception of the Mound Mix treatment, which had less than 70% similarity to other treatments (Table 2.1). The Mound Mix treatment had the highest number of native species followed by the intact and without organic, which had the same number of native species (Table 2.2). Although the Mound Mix treatment had the highest number of native species, it also contained the highest number of introduced species (Table 2.2). Soil weights and the number of stems per kg of soil were significantly different between treatments (F=4.193, d.f.=4,32 p=0.008; F=2.686, d.f.=4,32, p=0.049; respectively). A Dunnet's test resulted in the Mound Mix as the only treatment different from the control (Intact).

Twenty-nine species were identified and counted during the pre-move survey, and twenty-three species during the post-move survey (Table 2.3, Appendix B). There was no significant difference in the diversity of native species, but post-move diversity was significant lower than pre-move diversity for non-native species ($t_{(2,31)}$=0.63, p=0.530; $t_{(1,35)}$=-1.69, p=0.049; respectively). The density of native species stems increased marginally following the soil transfer, but was not a significant change ($t_{(2,15)}$=-0.87, p=0.397) (Figure 2.5). However, the density of introduced species stems decreased significantly ($t_{(2,15)}$=3.01, p=0.009) (Figure 2.5). A species accumulation curve indicated that sampling efforts were adequate for each mound as curves appear to reach asymptotes (Figure 2.6). The Sørensen index indicated that the pre-move comparison of mounds one and two shared only 50% of species in common and post-move comparison shared 72.7% of species. After the soil transfer, mound 1 pre- and post-move similarity was 31.6% whereas mound 2 pre- and post-move similarity was 25.6%, however after the soil transfer, the similarity was much greater (Table 2.4).

Discussion

A wider range of strategies gives land managers and property owners more tools which to apply management (Clout and Williams 2009). Using soil transfer accomplishes two objectives in maintaining a seed bank as well as reducing the amount of invasive species present at the recipient site (Brown 1998). The lack of studies on the effectiveness of soil transfer outside of a wetland has been addressed. This study provides supporting evidence that both of these can be accomplished, potentially saving time and money for land managers (Brown and Bedford 1997, Clout and Williamson 2009).

The greenhouse portion of this study indicated that all treatments contained a similar diversity of native species but invasive species diversity increased when the soil was mixed with soil from the recipient site. This is to be expected since the species are being added to the seed bank with this mixing process (Ter Heerdt et al. 1996). The Sørensen index supported this indicating a lower similarity between the Mound Mix and other treatments. Additionally, the Mound Mix treatment had the highest number of native species followed by the intact and without organic, which had the same number of native species. This would have been due to the addition of the natives which had established on the mounds through natural recruitment or through sowing efforts but had not established in the donor prairie (Ter Heerdt et al. 1996). Since the purpose of the soil transfer was to maintain the native seed bank while simultaneously controlling introduced species, the Intact or Without Organic treatments would have been the best way to transport the soil. However, these two transfer treatments are unrealistic; moving such a large area of soil fully intact is not feasible. While some grasses did occur in the greenhouse experiment, the vast majority of plants were forbs, adding to unrealistic nature of intact soil transfer via sod cuts at this site.

During the soil transfer, LRWP transferred the soil in a process similar to the Mix treatment used in the greenhouse portion, both in the process of moving the soil and in the similarity of diversity when looking at all species. This mixing was evident in personal observation of the soil pile created by the bulldozer, as well as subsequent loading and unloading of the dump truck. Natural recruitment was present in all of the trays as they contained species which were not present in the seed list provided by LRWP, both native and non-native species as seen in other research (e.g. Jakobsson and Eriksson 2003). The number of stems per kilogram was the highest in the Mound Mix treatment. Because the soil was mixed with both the seed bank from the donor and recipient site, the number of species present would be much greater (Brown and Bedford 1997).

The height of the first two mounds changed between the two surveys from the soil transfer (personal observation). I interpret the survey results as the soil transfer being successful in reducing the number of introduced stems while promoting the number of native stems. Diversity of the native species did not change while diversity of the introduced species was significantly reduced. The stems under the new layer of soil likely spent all of their stored resources in growth before reaching the surface, which allowed the soil transfer to be successful (Brown 1998, Reinhart and Callaway 2006). The soil transfer also increased the number of species shared between the mounds, lending further evidence to the successful movement of the seed bank. Although this study only consisted of one survey after the soil transfer, the species present in the seed bank included species which were naturally recruited (Jakobsson and Eriksson 2003).

The results of this study have demonstrated that transferring soil containing a native seed bank is an effective means to reducing introduced species density (Brown 1998). In this case, the soil transfer was cost effective since it preserved a native seed bank which would have been lost to future management activities (Clout and Williamson 2009). It was also effective in that no herbicides were applied to the mounds, which likely would have leeched into the neighboring wetlands (El-Sayed et al. 2006). Further studies may include a follow up survey to measure potential recruitment and preservation of the seed bank (Jakobsson and Eriksson 2003).

Table 1.1. Sørensen similarity index matrix (proportion of similar species) between greenhouse soil treatments.

	Intact	Invert	w/o Organic	Mix
Invert	0.750			
w/o Organic	0.769	0.811		
Mix	0.737	0.722	0.743	
Mound Mix	0.638	0.622	0.651	0.605

Table 1.2. Richness in greenhouse tray separated by treatment and species nativity status.

	Intact	Invert	w/o Organic	Mix	Mound Mix
Native	11	10	11	9	15
Introduced	10	9	7	8	10

Table 1.3. Richness of each mound from surveys conducted pre- and post-move separated by species nativity.

	Pre-Move		Post-Move	
	Mound 1	Mound 2	Mound 1	Mound 2
Native	10	16	15	10
Introduced	7	11	6	2

Table 1.4. Sørensen index matrix with proportion of similarities comparing pre-move mound species to post-move mound species.

	Pre-Move Mound 1	Post-Move Mound 2
Pre-Move Mound 2	0.500	0.256
Post-Move Mound 1	0.316	0.727

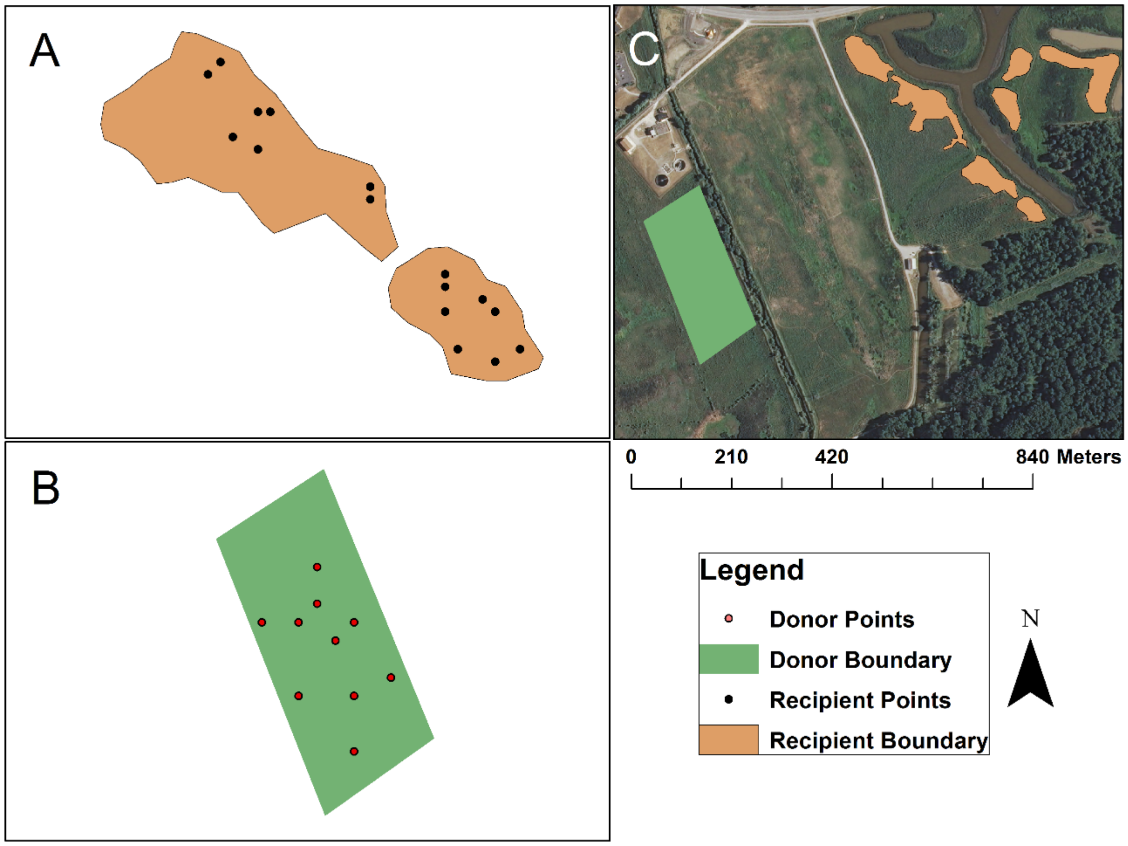

Figure 1.1. Detailed map of soil transfer (A) recipient area and survey points, (B) donor area and soil collection points, and (C) aerial base map with recipient and donor areas. Scale bar distance in reference to aerial map (C).

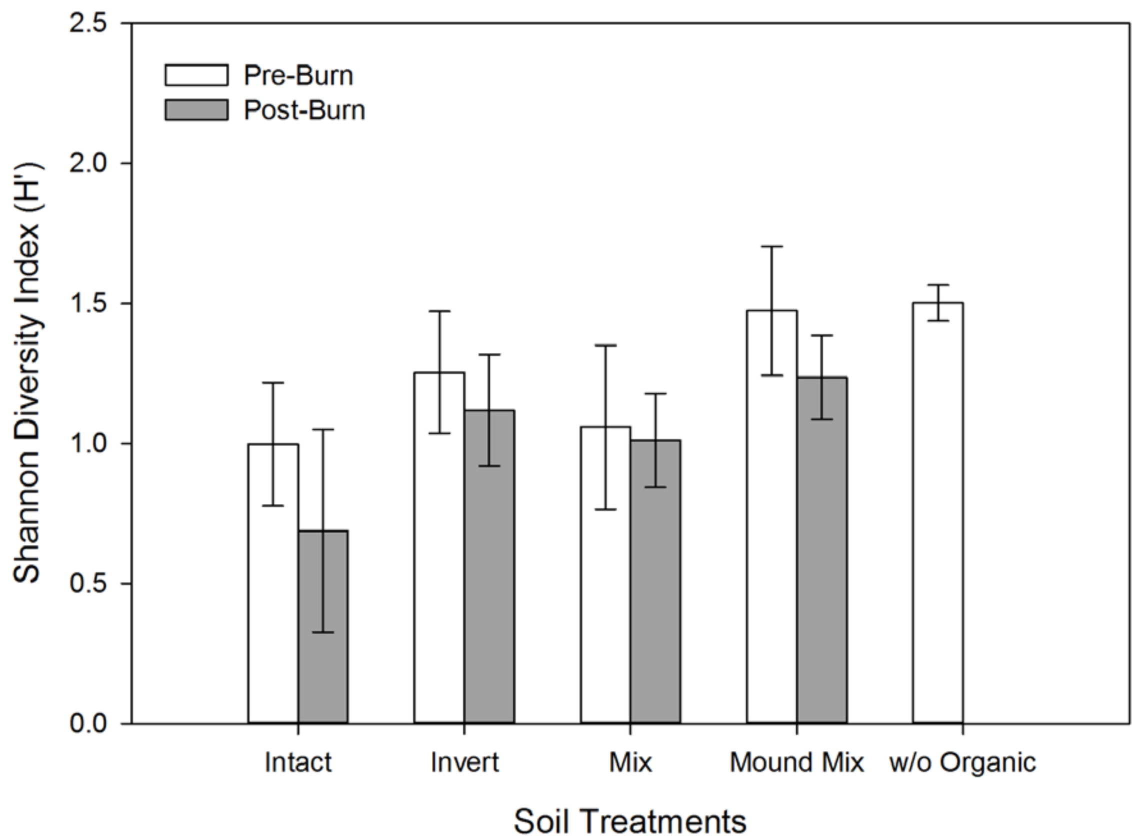

Figure 1.2. Native diversity for each of the treatments during the greenhouse portion of the experiment. White bars represent all trays collected before the control burn while grey bars represent those collected after.

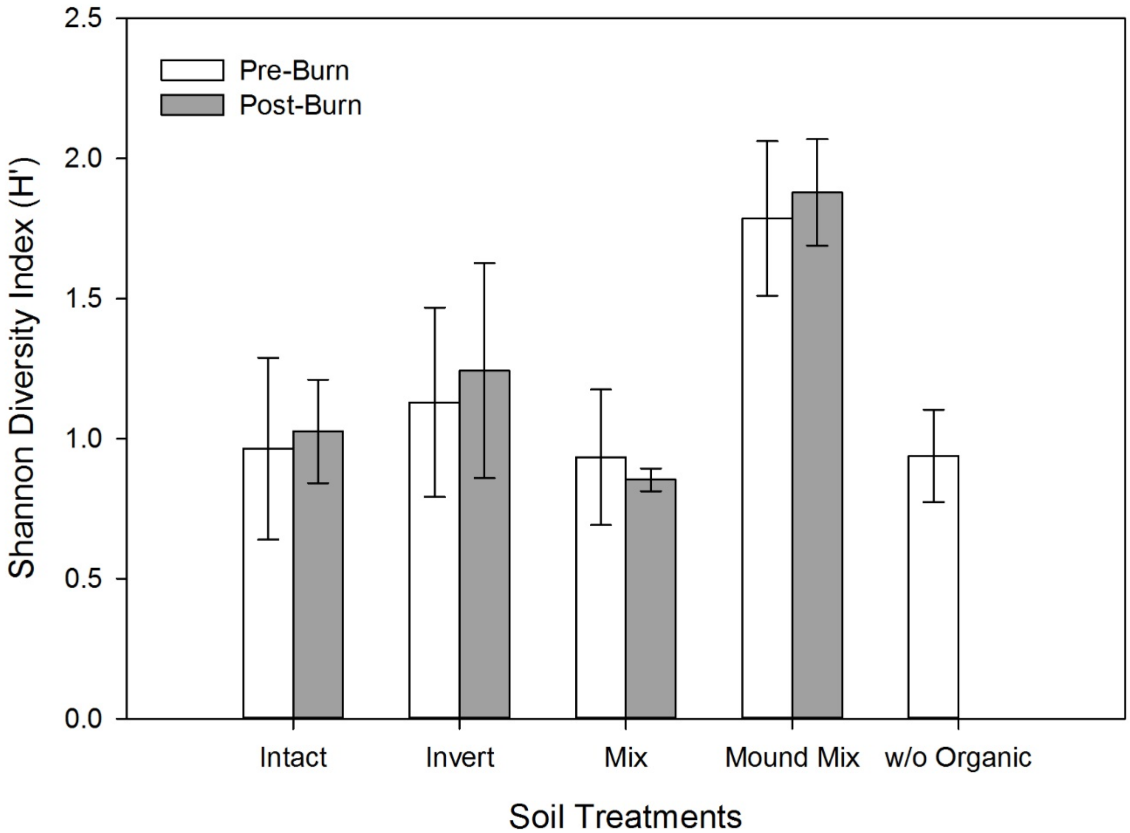

Figure 1.3. Introduced diversity for each of the treatments during the greenhouse portion of the experiment. White bars represent all trays collected before the control burn while grey bars represent those collected after.

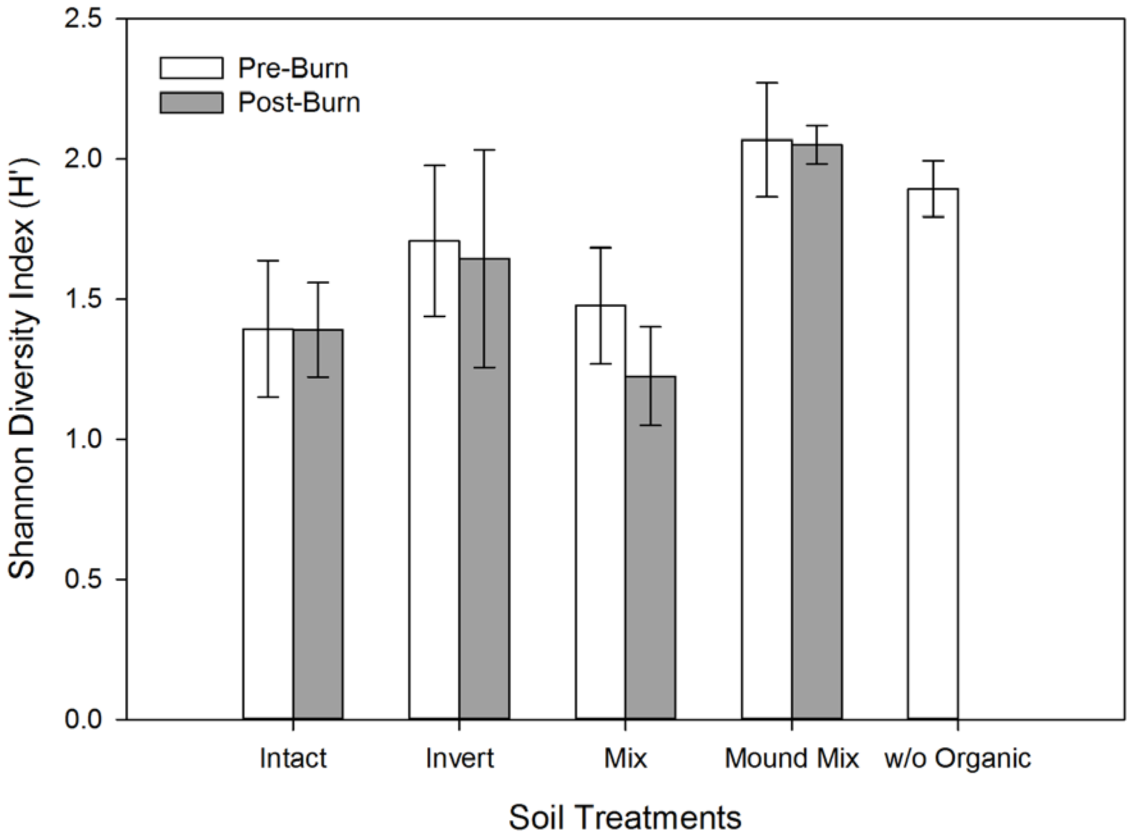

Figure 1.4. Pooled diversity of all species for each of the treatments during the greenhouse portion of the experiment. White bars represent all trays collected before the control burn while grey bars represent those collected after.

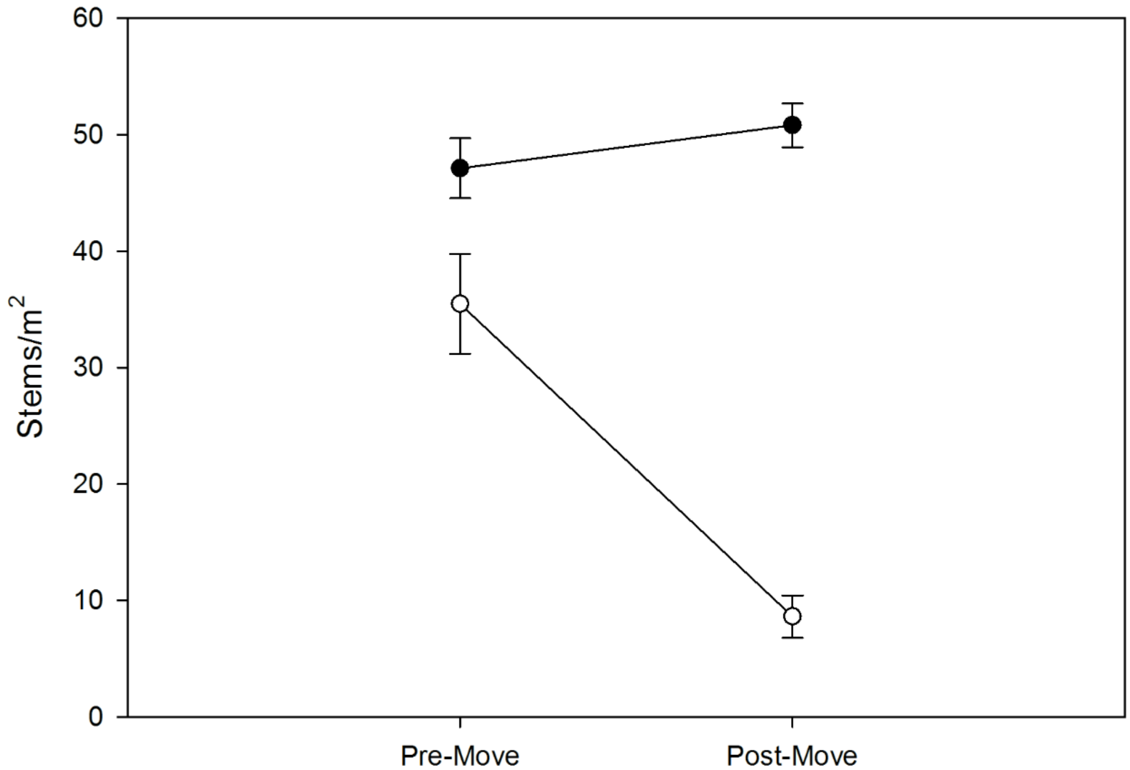

Figure 1.5. Density of stems pre- and post-move separated by nativity with standard error. Closed circles represent native species while open circles represent introduced species.

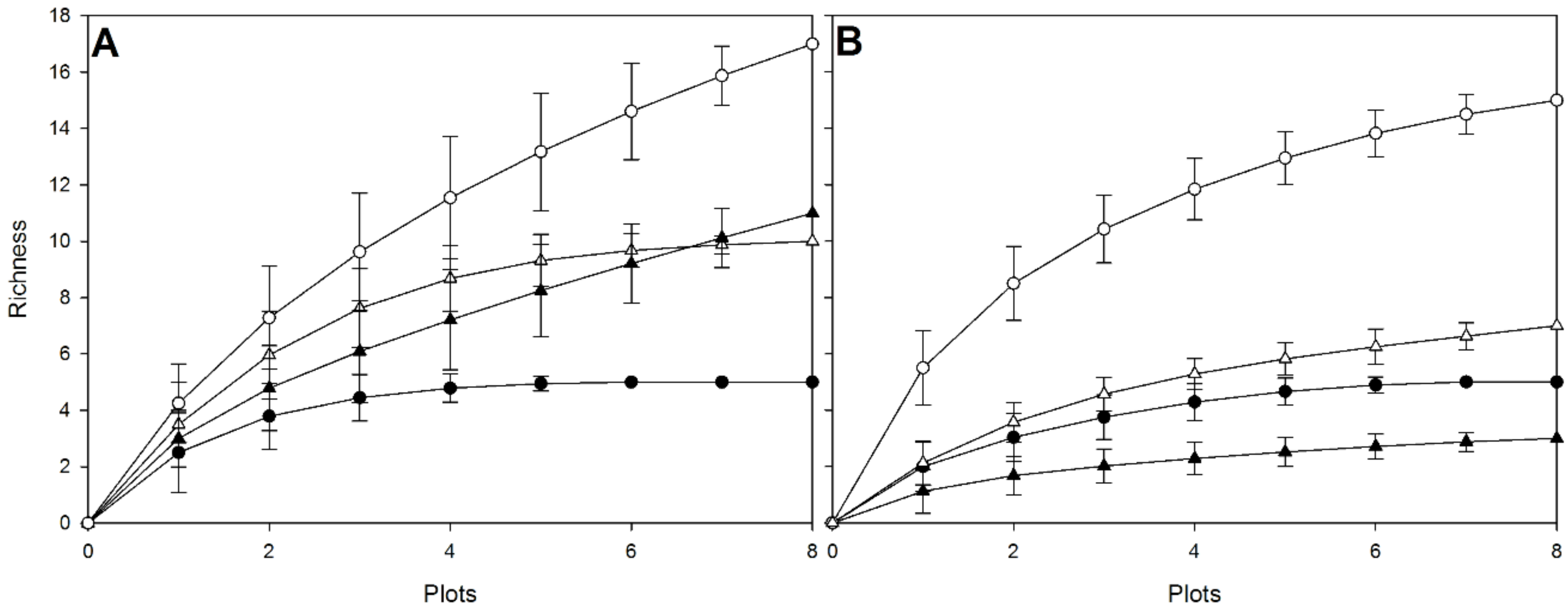

Figure 1.6. Species accumulation curves for recipient mounds associated with survey conducted in (A) July 2014 (pre-move) and (B) August 2015 (post-move). Open symbols represent native species and closed symbols represent introduced species. Circles represent mound one and triangles represent mound two.

CHAPTER 2. INFLUENCE OF FIRE ON CALLERY PEAR (*PYRUS CALLERYANA*) SURVIVAL AND RECRUITMENT IN A MANAGED PRAIRIE ECOSYSTEM

Introduction

Callery pear (*Pyrus calleryana* Decne. [Rosaceae]) is an introduced tree in North America originally from Asia and widely planted as an ornamental, yet little research has been conducted on the ecological or economic impacts of the introduction (White et al. 2005). It was originally introduced due to its resistance to fire blight, a bacterial disease, and became a common ornamental tree in the 1950's (Reimer 1925, Creech 1973). It has spread through the eastern United States and establishes in prairies, old fields, and other disturbed areas (Vincent 2005, Culley and Hardiman 2007). However, as distance from roads increases into forests, Callery pear density decreases (Flory and Clay 2006). Vincent (2005) found that Callery pear exhibits improved establishment and recruitment in mid- to late-successional habitats.

Management of invasive species can be challenging since means of control may be difficult to implement. Culley and Hardiman (2007) have demonstrated the effectiveness of hand removal techniques, but noted that these are labor intensive. Even though there has been past evidence of Callery pear in prairies, fire has not been studied as a suitable management strategy for control.

There have been few studies on the ecological effects of Callery pear, although it has been predicted that they can impede establishment of late to middle stage successional species (Culley and Hardiman 2007). Thorny thickets can form causing a management problem if the thorny wild-type is present. Fulcher (2002) noted that many Callery pear trees planted in urban environments are susceptible to limb breakage. This combined with the litter of the fruit on the ground can cause both an unpleasing aesthetic effect and a potential danger to fool traffic (Dirr 1998, Fulcher 2002). Even with such few studies of impacts, Callery pear has been listed by Indiana DNR as invasive in natural areas and as a “high” rank by the Nature Conservancy and Indiana Invasive Species Council (Homoya 2010, Jacquart 2012, IISC 2013).

Colonization by Callery pear into prairies leads to interactions between common disturbances and this invasive species. Fire is a common ecological disturbance in many ecosystems (Neary et al. 1999). Plant biomass consumed by fire can be living or dead although most of the released carbon comes from dead plant litter (Raison et al. 1985). The irregularity of fire contributes to the effect that it has on an ecosystem, with variability in frequency, seasonality, and intensity (DeBano et al. 1998). These three factors combine to determine fire regimes, which are used to create management strategies in different ecosystems (Whitlock et al. 2010). Lightning is the most common ignition source of grassland fires (Renkin & Despain 1992). Many grasses produce seeds that rely on fire, which serves to release them from dormancy and to prepare the soil for the seeds by altering resources by reducing competition (Ojima et al. 1994, Auld and Bradstock 2006). Prairies can be colonized by trees, but fire typically keeps these areas open and disturbed to allow for herbaceous, prairie species to dominate (Brooks et al. 2004).

Due to the lack of studies in the effectiveness of fire on Callery pear, an appropriate experiment is necessary. This had led to the current study. The objectives of this study were to 1) map the distribution of Callery pear individuals within Arrowhead Prairie, Allen County, Indiana; 2) quantify the growth response of Callery pear following prescribed fire; and 3) test the hypothesis that fire is an acceptable management strategy for controlling Callery pear invasions.

Materials and Methods

Site Description

Arrowhead Prairie is a 64 hectare prairie located in southwest Allen County, Indiana (16N 4540538 641389, Figure 3.1), with 27 hectares being used for this study. Prior to acquisition by Little River Wetland Project (LRWP) in 2009, the property was used for row crop agriculture. The prairie was divided into two main sections by the historic Graham McCulloch Ditch, which runs from the west central section of the prairie to the southeast corner (Figure 3.2). The ditch was previously fed by the Aboite Creek and a section of the ditch which connects Aboite Creek to Arrowhead Prairie passes through agricultural land and is currently filled with soil. Historic aerial photos show that from the 1930s to the 1960s the ditch was still connected to Aboite Creek and was lined trees along the edge (IHAPI 2013a,b,c). Two ponds are located in the southern section and were constructed after LRWP acquired the property.

Data Collection

A 20 meter spaced grid was layered over an aerial photograph of Arrowhead Prairie using ArcGIS (ESRI, Redlands, CA, version 10.2.2). The prairie was divided into the north and south sections using the historic Graham McCulloch Ditch as the division, serving as a major fire break for the subsequent burns (Figure 3.2). From each section, fifteen points were randomly selected and a five-meter radius circular plot was established around each point. All Callery pear trees within each plot were measured for demographics, which included tree height, root collar diameter, distance and azimuth from plot center, and damage to the tree (rabbit and deer feeding damage).

Prescribed burns were applied to the southern section of the prairie on 16 April 2014 and to the northern section on 7 May 2015. South pre-fire demographics were collected in April 2014, south post-fire demographics were collected on 23 July 2014. North pre-fire demographics were collected in November 2014 and March 2015 (prior to 2015 bud break) and north post-fire demographics were collected on 20 July 2015. The number of epicormic sprouts were recorded in post-fire surveys. Due to continuous, active management of Callery pear by LRWP, the study concluded after post-fire 2015 data collection as the majority of trees were lost to cutting and herbicide application.

To measure fire temperature during the 2015 burn, ceramic tiles with temperature indicating lacquers were placed at 95 points on the 20 m spaced grid. Bamboo sticks were used to hold two 10 cm x 10 cm tiles lacquer face down approximately 50 cm above the soil surface. Each tile was painted with temperature indicating lacquers ranging from 93 °C to 538 °C. Tiles were removed post-fire, and the maximum temperature indicating lacquer that melted was recorded.

Prairie biomass was sampled in 0.5 m x 1 m quadrats at 10 locations in the prairie during the north pre-fire demographics survey in March 2015. Biomass samples were collected at randomly selected mid-points between the 20 m spaced grid points to ensure biomass collections did not interfere with fire temperature measurements where tiles were located. Biomass samples were dried in an oven at 50 °C to a constant mass. Samples were returned to the prairie before the 7 May 2015 fire. In order to further describe the site, a rapid survey of the prairie plant species was conducted on 21 August 2015. Percent cover by species within a 1 m^2 quadrat was estimated in stochastically located plots. This survey was not intended to be an exhaustive survey of the plant community present.

Analysis

A Student's t-test was used to determine differences between root collar diameter and height between years and treatments. A chi-squared test was performed on animal foraging damage to trees and the number of sprouts produced by trees. Density of Callery pear trees in the prairie sections were compared using a blocked two-way ANOVA. All statistical analyses was conducted in R using R Commander (Fox 2005, R Core Team 2015). Fire temperature and biomass were mapped using Inverse Distance Weighting (IDW) interpolation with modeling extended to prairie boundaries (ESRI, Redlands, CA, version 10.2.2).

Results

The northern section of the prairie contained areas of lower elevation, which flooded during the survey. Conversely, the southern section was higher in elevation due to additional soil likely accumulated from the building of the two ponds. The dominate plant species was *Spartina pectinata* Bosc ex Link (Poaceae) with *Cirsium arvense* (L.) Scop. (Asteraceae), *Sorghastrum nutans* (L.) Nash (Poaceae), and *Doellingeria umbellate* (Mill.) Nees (Asteraceae) being common species. *S. pectinata* comprised 72% of the total cover of plots measured, which was an order of magnitude higher than *C. arvense,* which had 7.1% total cover.

The trees in the south prairie were significantly larger in both height and root collar diameter when compared to trees in the north prairie (height: $t_{(1),305} = -4.69$, $P < 0.001$; root collar: $t_{(1),305} = -3.90$, $P < 0.001$). Density of the trees on the northern section of prairie was much lower compared to the southern section of prairie ($F=28.26$, d.f.=1,112, $P<0.001$). Mean density of trees in the northern prairie was 44 trees per hectare compared to 209 trees per hectare in the southern prairie. There was no significant difference in density based on the time of the survey or the interaction of the time and prairie section ($F=0.97$, d.f.=3,112 $P=0.411$; $F=0.257$, d.f.=3,112 $P=0.856$; respectively).

After the application of fire to the south prairie, all measured trees were top killed. Post-fire 2014 included epicormic sprout counts, which all measured trees produced at a mean of 3.1 per tree (SD=1.8). The trees were significantly shorter after the fire compared to early-season. Root collar diameter was not significantly different after the fire (Table 3.1). Similarly, in the north prairie all trees were top-killed and subsequently produced basal epicormic sprouts with a mean of 4.1 sprouts per tree (SD=1.6). Additionally, trees were significantly shorter after the fire and root collar diameter was not significantly different (Table 3.1). Pooled across north and south sections of the prairie, epicormic sprouting was independent of foraging damage by deer and rabbits (X^2<0.001, d.f.=1, P=1.000; X^2=0.16, d.f.=1, P=1.000; respectively).

The average maximum temperature of the fire, measured in the north prairie, was 195 °C (range: 121-253 °C; Figure 3.3). The highest temperatures were recorded near the eastern edge, southern edge, and center of the burn area (Figure 3.4). The lowest temperatures were recorded along the western edge (Figure 3.4). Analysis of biomass using IDW indicated a higher biomass measurement along the eastern edge, southern edge, and center, with lower biomass measurements along the eastern edge (Figure 3.5).

Discussion

Callery pear has the potential of becoming one of the most problematic invasive species in the United States (Culley and Hardiman 2007). There is a dearth of knowledge on the ecological impacts of this invader and should be studied in order to provide a better understanding into management strategies which may help in the reduction of those potential ecological impacts (Vincent 2005). I attempted to use a management strategy which was already used in prairies as a viable method to control Callery pear (Vincent 2005). This could have potentially reduced both the monetary and labor costs of effective Callery pear management (Culley and Hardiman 2005).

The results presented here indicate that fire was not a viable technique for controlling Callery pear. While the vast majority of trees were top-killed after the prescribed burns, all measured trees had epicormic sprouts resulting in increased numbers of stems. This increase in stems could present a problem from a management standpoint due to the likely subsequent increase in flower and fruit production (Kauffmann 1991, Bond and Midgley 2001). Even though none of the Callery pear trees surveyed at Arrowhead Prairie produced flowers, an increase in the number of flower producing stems could cause an increase in the number of tress in the prairie (Gonzalez et al. 2015). Fire did top kill all of the trees, which would have destroyed any flower buds on the stems (Bond and Midgley 2001). Potentially, fire may have artificially reduced the age of trees by forcing regrowth and delaying flower production (Kauffmann 1991, Gonzalez et al. 2015). Trees measured after the burn were significantly shorter than trees measured before the burn but the root collar diameter did not change. Since the trees needed to regrow stems in a single growing season, it isn’t surprising that the post-fire trees were shorter than pre-fire. The trees most likely put more resources into primary growth rather than secondary growth (Nowak and Crane 2002, Kennard et al. 2002).

Fire temperature IDW aligned well visually with biomass IDW, increased temperature with an increased biomass. This was expected since there was more fuel to consume. Mean recorded temperature was close to average temperatures of typical grassland fires in the Great Plains indicating that the Arrowhead Prairie burn should be considered a normal prairie burn (Lata and Weirich 1999). An increase in the temperature may kill Callery pear trees but may be problematic from a management stance of adding more fuel for the fire to consume, controlling the larger fire, and causing damage to the existing plant and animal community present in the prairie (Hulbert 1988, Davidson and

Janssens 2006). There would also be a greater release of CO_2 which would not be typical of a prairie given the plant community and fire regime (Davidson and Janssens 2006).

Animal damage combined with fire did not influence the number of basal epicormic sprouts or height of the trees. During 2014, 13% of trees in the northern prairie had basal epicormic sprouts without fire. Since these trees were not exposed to fire until the 2015 burn, animal damage is thought to be the main cause of these sprouts although other underlying factors may be present (Wenger 1953, Kennard et al. 2002).

Density of trees in the northern prairie was much lower than in the southern prairie. The southern prairie is lined by trees on the western edge, across a county road from a forested area to the east, and adjacent to landscaping trees to the south. The northern prairie is near the western tree line, but mostly is not close to other trees or forest. Birds who have consumed the Callery pear fruit may have flown over the south prairie and dispersed the seeds, resulting in a greater density of trees in the south compared to the north (Levey et al. 2005).

Using a cutting and herbicide application, Callery pear can be controlled (Culley and Hardiman 2005). Given that fire reduces the amount of biomass in the prairie, an herbicide application may be more practical due to accessibility after a controlled burn (DeBano et al. 2008). Finding a native animal species to serve as a biocontrol agent may aid in the reduction stem density (Kennard et al. 2002). Further studies should aim to look at the factors allowing for the spread of Callery pear and the ecological impacts of the spread. Further public education of Callery pear as an ornamental species may assist in a reduction of intentional plantings, given the amount of native alternatives which can be planted (Jones 2004).

Table 2.1. Mean Callery pear tree height and root collar diameter pre- and post-burn at Arrowhead Prairie (standard error) within north and south halves of the prairie. Significant p-values are indicated with an asterisk (*).

Characteristic	**Pre-Burn**	**Post-Burn**	**t**	**df**	**p-value**
South Height	104.82 (26.34)	92.53 (18.67)	6.20	550	<0.001*
South Root Collar	16.03 (7.41)	18.14 (15.72)	-1.80	476	0.073
North Height	115.32 (24.56)	71.68 (13.69)	9.19	71	<0.001*
North Root Collar	21.16 (7.63)	21.31 (9.30)	-0.07	71	0.94

Figure 2.1. Map detailing the location of Arrowhead Prairie within Allen County with reference to Fort Wayne Metropolitan Area. Insert shows location of prairie in reference to state.

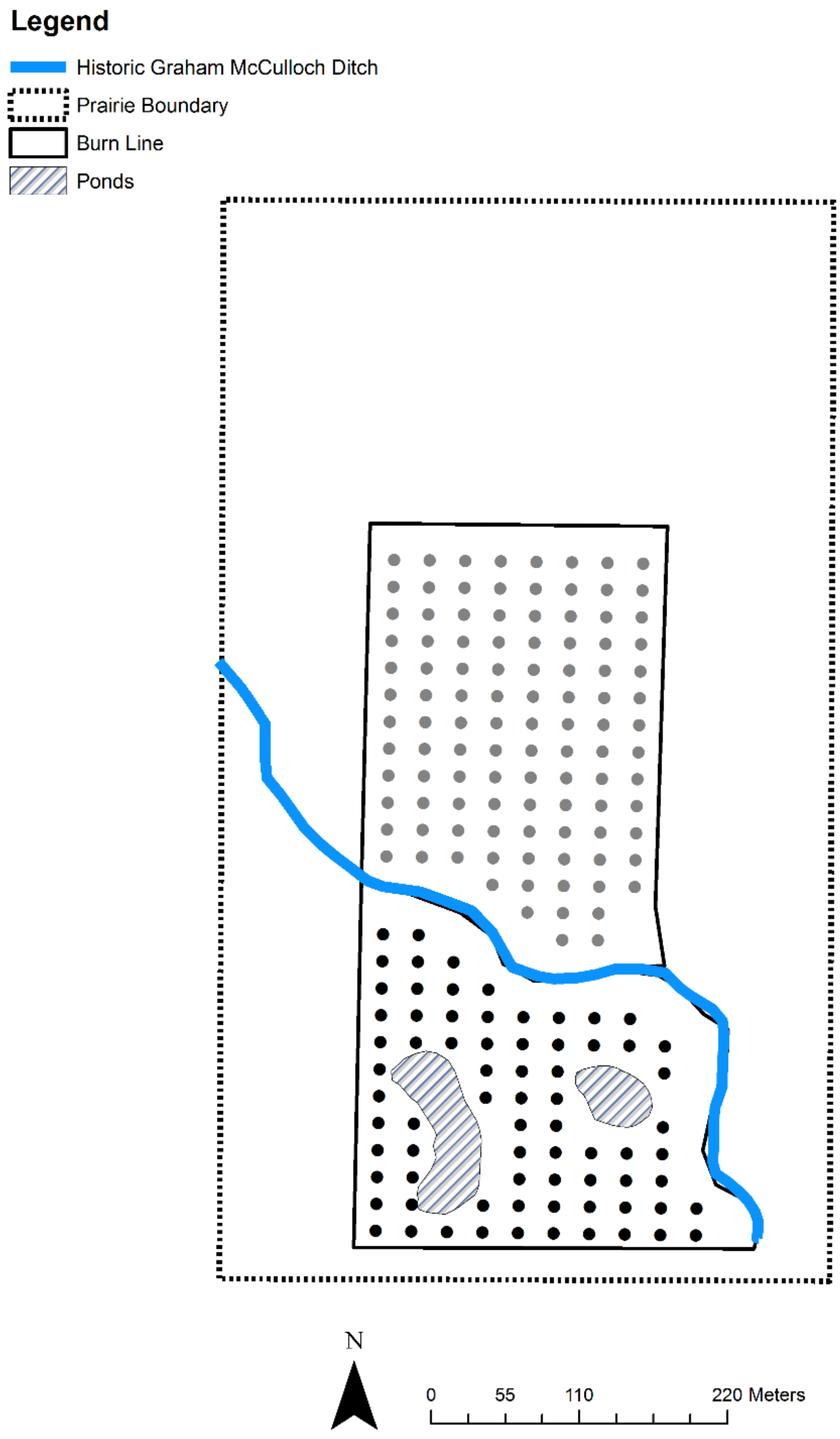

Figure 2.2. Map detailing Arrowhead Prairie division using Historic Graham McCulloch Ditch and point grid used for sampling. Note: northern boundary of the entire prairie extends 240 m north of the burn boundary.

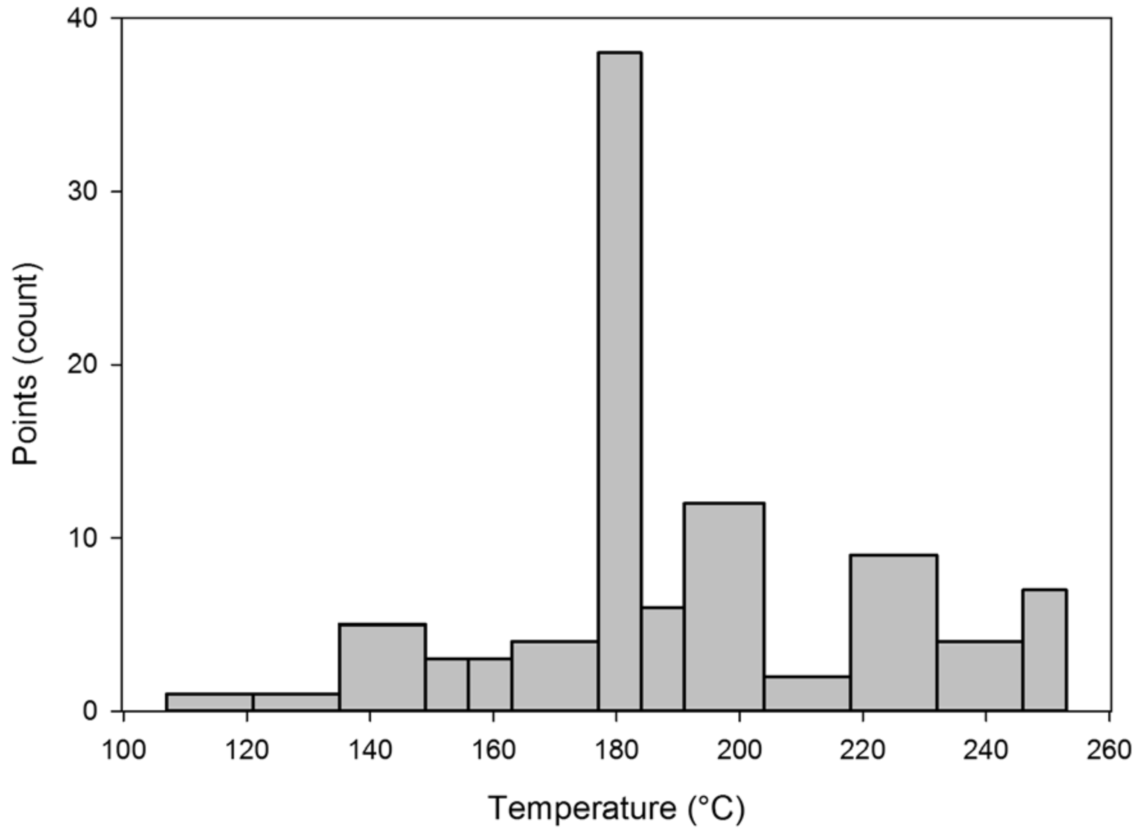

Figure 2.3. Histogram of sampling points with corresponding temperature from the prescribed burn. Right edge of bar indicated the highest temperature measured by each lacquer. Wide bars represent lacquers that covered a large temperature range.

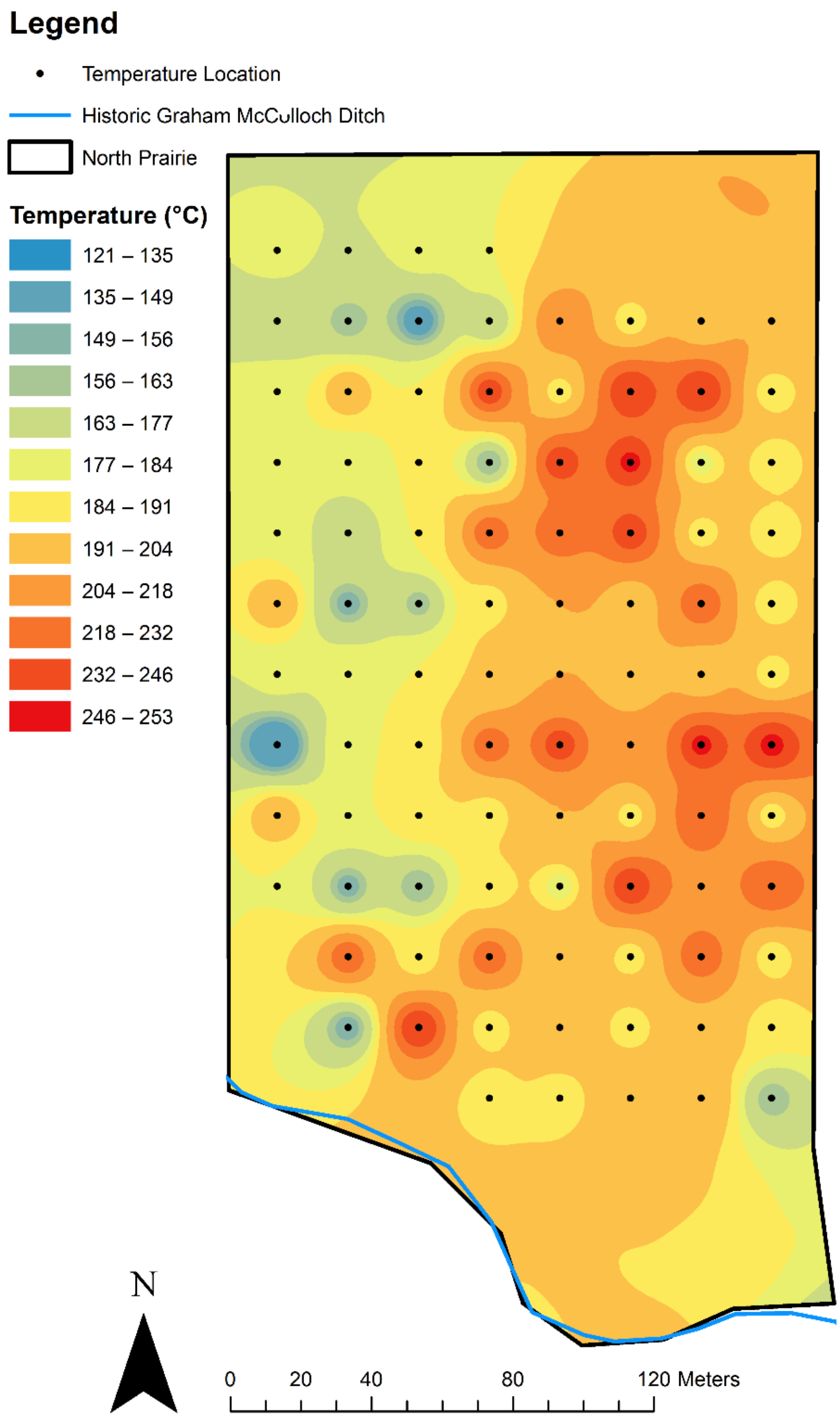

Figure 2.4. Inverse distance weighing (IDW) interpolation representing maximum temperature across Arrowhead Prairie.

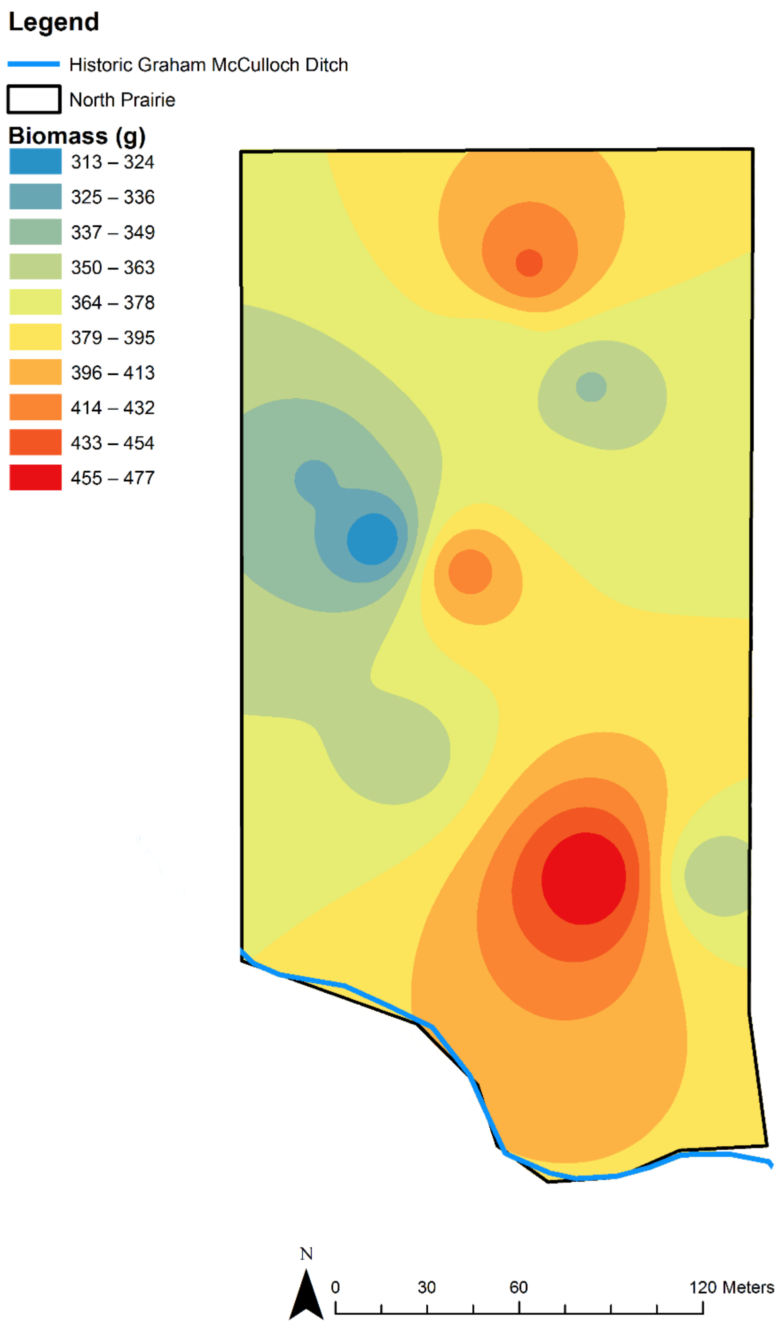

Figure 2.5. Inverse distance weighting (IDW) interpolation representing dry biomass weight across Arrowhead Prairie.

CHAPTER 3. FIRE AND ITS INFLUENCE ON CALLERY PEAR (*PYRUS CALLERYANA*) FRUITS, SEEDS, AND SEEDLINGS AFTER A CONTROLLED BURN

Introduction

Fire is a common disturbance which various plants have adapted to as part of their life cycle, yet, fire can also be used as a mean of control for introduced species (Clout and Williamson 2009). Although an attempt was made to control the spread of Callery pear (*Pyrus calleryana* Decne. [Rosaceae]) using fire, the study was limited to established saplings and did not include fruit and seed mortality(Chapter 3). Callery pear exhibited an ability to resprout after fire (Chapter 3) but this same resilience may not be present in the fruit or seeds (Keeley 1977, Lamot and Wiens 2003). Given the high rate of dispersal by birds, managing the germination rate of fruit and seeds may help in controlling the growth and spread of Callery pear (Stiles 1980, Culley and Hardiman 2005). The objective of this study was to quantify the mortality rate of fruit, seeds, first-year seedlings, and second-year seedlings when exposed to fire.

Materials and Methods

Fruit were collected from two mature Callery pear trees on IPFW campus (UTM 16N 4553607 658772). Seeds were removed from a random sample of fruits. Three hundred seeds were subjected to fire and placed in cold storage 3 November 2014. One hundred whole fruits were subjected to fire and placed in cold storage 15 November 2014. These seeds and fruit would serve as the pre-winter burn treatment group. Six hundred forty-nine seeds and two hundred whole fruits were placed in cold storage 31 October 2014 to serve as the control. Cold storage was conducted at 5 °C with wet stratification. All seeds were removed from cold storage on 26 February 2015 due to the seeds germinating in cold storage. After the seeds were removed from cold storage, 186 seeds and 100 fruit were subjected to fire on 28 February 2015. These would serve as the post-winter burn treatment group. The remaining 463 seeds and 100 fruit were used as a control.

Ninety-eight germinated seeds were planted into greenhouse trays and stored in a greenhouse and maintained at 21 °C and approximately 33 percent relative humidity, with ambient light. Day length was extended to 18 hours with sodium vapor lamps. Of the 98 trees which germinated in cold storage, 34 failed to establish in the greenhouse. The remaining 64 trees which established were divided into two trays based on leaf count with each tray received 32 trees. One tray, represented first-year seedlings, was burned on 6 May 2015. The second tray, representing second-year seedlings, was burned on 7 Jan 2016. Trays were returned to the greenhouse after burn to allow for germination. After three months, the trays were checked for sprouting. Fire temperatures were measured using a 10 cm x 10 cm ceramic tile containing temperature sensitive lacquers positioned approximately 50 cm above the soil surface. Biomass from Arrowhead Prairie was used as the fuel for the fire with an average depth of 25 cm.

Results

While in cold storage, 98 seeds from the control group and 114 seeds from the post-winter fire group germinated. The fire on the pre-winter and post-winter burn fruits and seeds measured at 149 °C. None of the seeds germinated that were exposed to fire as bare seeds or contained in fruits. First-year seedlings were subjected to a fire of 156 °C. After three months, one sapling produced epicormic sprouts. Second-year seedlings sustained a fire of 184 °C. All 32 trees produced epicormic sprouts (Figure 4.1).

Discussion

Management techniques which aim to control the spread of fruit and seeds have shown to be effective yet difficult to implement (Clout and Williamson 2009). Control of Callery pear can be problematic given that once the trees are established, they are difficult to control (Culley and Hardiman 2007, Chapter 3). Callery pear has a remarkable ability to store resources in their roots which allow for resprouting after fire (Chapter 3). Callery pear seeds were able to germinate when in cold storage at 5 °C demonstrating that the application of growing degree-days may not be appropriate for estimating germination (McMaster and Wilhelm 1997, Herms 2004). Fire caused a 100% mortality seeds exposed bare and contained in the fruit. First-year seedlings had a 3% resprout rate yet the second-year seedlings were able to store appropriate resources in the roots to enable all trees to produce epicormic sprouts (Nowak and Crane 2002). The 3% resprout rate on the first year seedlings (one single individual) may be due to a phenotypic growth trait of the specific cultivar where that individual may have grown fast enough to store suitable resources for resprouting (Atkins et al. 2008, Murphy et al. 2008). Cutting and spraying may be the most practical method of controlling Callery pear once it reaches second-year growth (Culley and Hardiman 2007). Reduction in the number of plantings will reduce the amount of trees which can provide a seed source (Culley and Hardiman 2009). Callery pear is incompatible with itself but is capable of hybridization which adds a greater emphasis to the reduction in populations to control the spread of fruit and seeds (Culley and Hardiman 2007, 2009). If seeds are dispersed, control burns are capable of controlling the germination of trees until the trees reach their second year.

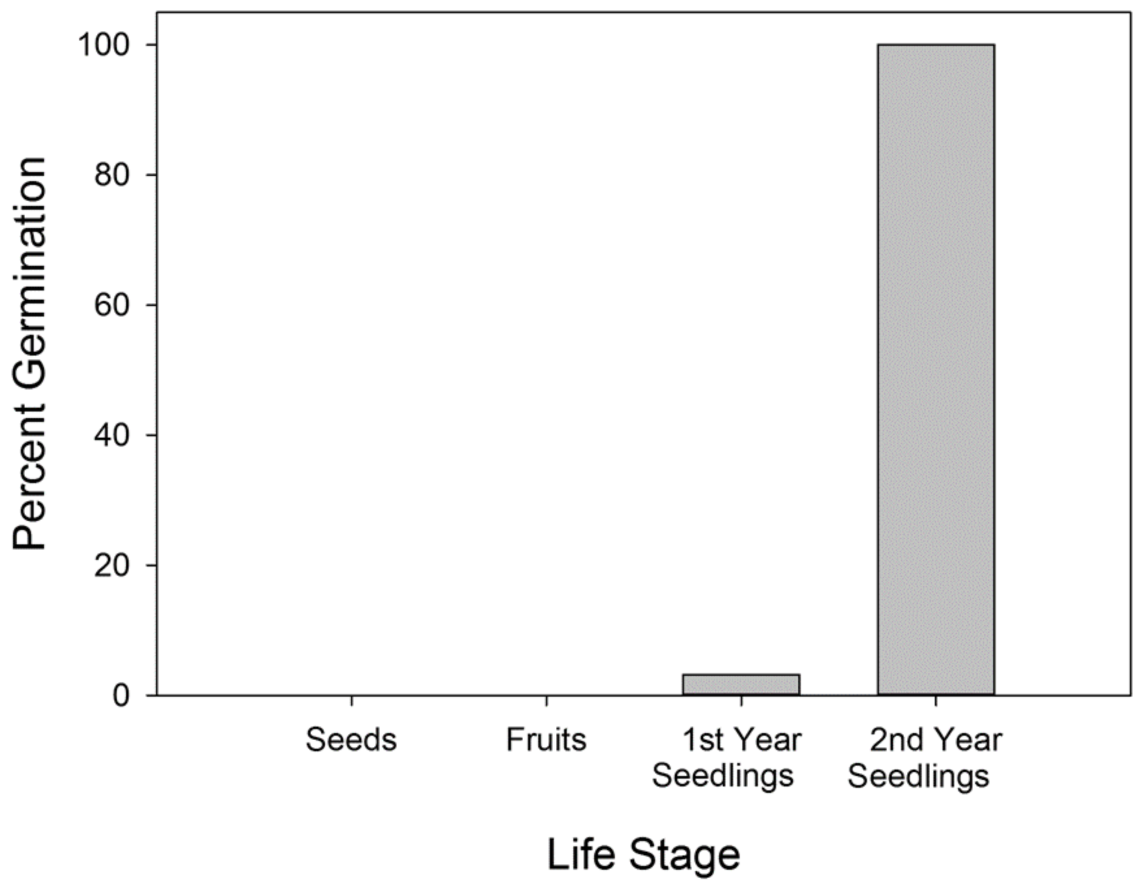

Figure 3.1. Proportion of germination of seeds exposed to fire as bare seeds and contained in fruits and resprouts from first and second year seedlings.

CONCLUSIONS

The overall objective of my thesis was to test the effectiveness of two management strategies (soil transfer and prescribed fire) on controlling invasive species within specific management scenarios. These two strategies have very different management objectives. The intention of soil transfer in this case was to preserve a native seed bank, promote native species, and diminish abundance of invasive species. The intention of prescribed fire was to reduce standing biomass, promote native species, and reduce the density of a specific invasive species (Callery pear).

Soil transfer was shown to be successful in maintaining the seed bank but also reducing the number of invasive species located on the mounds. This is a benefit to LRWP in that they saved the seed bank in which they invested time and money but also accomplished the goal of reducing the number of invasive species on the mounds. The transferred soil did contain invasive species, admitted a lower diversity and density, so continuous management is still necessary if there is a desire to further reduce the amount of invasive species.

The donor soil contained seeds which LRWP sowed. Not all of these plants were present during the greenhouse portion of the study. The absence of those plants along with the natural recruitment of other native and invasive species suggests that the mound species composition may also change. The soil composition under the transfer and the new location of the soil may cause a change in species composition. Studies have suggested that the composition of the seed bank is a poor predictor of the plant community after transfer yet these studies were conducted in wetlands, not on prairie mounds. A follow up survey will be necessary to measure the persistence of the seed bank as well as recruitment of new species.

While the soil transfer was effective, fire did not demonstrate the same effectiveness in controlling Callery pear. Callery pear responded to fire by producing epicormic sprouts. This emphasizes the amount of resources stored in the roots of the tree. Culley and Hardiman (2007) discussed the importance of management but also the hardiness of Callery pear. Given the response to fire, cutting and herbicide application may be the best approach when attempting to remove Callery pear.

Fire did, however, have a significant effect of the mortality of fruit, seeds, and first-year seedlings. Although fire did not reduce the number of trees in the prairie, it may be helpful in preventing future recruitment. Controlled burns to increase seed and first-year seedling mortality combined with cutting and herbicide application would result in a decrease in the number of trees in the prairie. Continuous management is needed to make this a possibility.

Callery pear was not the only invasive species in the prairie. The second most abundant species in the prairie was Canada thistle. Removal of Canada thistle can be achieved along with removal of Callery pear. As Culley and Hardiman (2007) pointed out, hand removal of invasive species can be labor intensive. This can be reduced after a controlled burn when built-up biomass and fuel have been removed and the current year’s growth is short.

Little River Wetlands Project has the information from this thesis and modified the management of their property. The soil transfer did move invasive species to the mound. LRWP has decided to focus on the more persistent and problematic species, like Canada thistle (*Cirsium arvense* (L.) Scop. [Asteraceae]). Management at Arrowhead Prairie has involved the removal on Callery pear by cutting and herbicide application. While Callery pear is a problem at Arrowhead Prairie, other invasive species are present, although they were not the focus of the Callery pear study. Also, Callery pear invades property after a disturbance so particular attention should be paid to sprouts after a prescribed burn.

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APPENDICES

Appendix A Annotated Species List

List of species encountered in soil treatments during Chapter 2 greenhouse experiment. Species binomial followed by soil treatment in which the species occurred: Intact – an intact soil sample with organic litter; Invert – an intact soil sample which was completely inverted; w/o Organic – an intact soil sample with the organic litter removed by hand; Mix – a soil sample mixed homogenously by hand; Mound Mix – a soil sample mixed homogenously by hand with an equal amount of soil from the recipient site; Treatment w/ Fire – a soil sample handled similar to treatment name but collected after the controlled burn. Nomenclature follows ITIS (2016). An asterisk (*) indicates non-native species (USDA NRCS 2016). A dagger (†) indicates species that were included on the Little River Wetlands Project seed list.

AMARANTHACEAE

* *Chenopodium album* L.: Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mix w/ Fire, Mound Mix w/ Fire

* *Chenopodium murale* L.: Mix, Mix w/ Fire, Mound Mix w/ Fire

Chenopodium standleyanum Aellen: Mound Mix, Mound Mix w/ Fire

APIACEAE

* *Daucus carota* L.: Intact, Invert, w/o Organic, Mix, Mound Mix

APOCYNACEAE

† *Asclepias incarnata* L.: Intact w/ Fire

ASTERACEAE

* *Ambrosia trifida* L.: Intact, Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mound Mix w/ Fire

* *Cirsium arvense* (L.) Scop.: Intact, Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mix w/ Fire, Mound Mix w/ Fire

* *Lactuca serriola* L.: Intact

* *Leucanthemum vulgare* Lam.: Intact, Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mix w/ Fire, Mound Mix w/ Fire

Packera glabella (Poir.) C. Jeffery: Invert, Invert w/ Fire

† *Rudbeckia fulgida* Aiton: Mound Mix, Invert w/ Fire

† *Rudbeckia hirta* L.: Intact, Invert, w/o Organic, Invert w/ Fire

† *Silphium perfoliatum* L.: Intact

* *Sonchus oleraceus* L.: Intact, Invert, w/o Organic, Mound Mix

Symphyotrichum dumosum (L.) G.L. Nesom: Intact, w/o Organic, Mix

† *Symphyotrichum puniceum* (L.) Á. Löve & D. Löve: Intact, Invert, w/o Organic, Mix, Mound Mix

* *Taraxacum officinale* F.H. Wigg.: Intact, Invert

† *Vernonia gigantea* (Walter) Trel.: Intact, w/o Organic, Mix, Mound Mix, Invert w/ Fire

BRASSICACEAE

Rorippa palustris (L.) Besser: Mound Mix

* *Thlaspi arvense* L.: Mound Mix, Mound Mix w/ Fire

CYPERACEAE

† *Carex hystericina* Muhl. ex Willd.: Intact, Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mix w/ Fire, Mound Mix w/ Fire

CUPRESSACEAE

Juniperus virginiana L.: Mix

FABACEAE

† *Senna hebecarpa* (Fernald) H.S. Irwin & Barneby: Invert

LAMIACEAE

† *Monarda fistulosa* L.: Intact, Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mix w/ Fire, Mound Mix w/ Fire

MALVACEAE

* *Abutilon theophrasti* Medik.: Intact, Invert, Mix, Mound Mix, Mix w/ Fire, Mound Mix w/ Fire

* *Hibiscus trionum* L.: Intact w/ Fire

MOLLUGINACEAE

Mollugo verticillata L.: Mound Mix, Mound Mix w/ Fire

ONAGRACEAE

Ludwigia palustris (L.) Elliott: Mound Mix

PHYTOLACCACEAE

Phytolacca americana L.: Mix w/ Fire

PLANTAGINACEAE

† *Penstemon digitalis* Nutt. ex Sims: Intact, Invert, w/o Organic, Mix Mound Mix, Intact w/ Fire, Invert w/ Fire, Mound Mix w/ Fire

POACEAE

† *Andropogon gerardii* Vitman: Intact, Mix, Mound Mix

† *Panicum virgatum* L.: w/o Organic, Mound Mix

* *Setaria faberi* R.A.W. Herrm: Intact, Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mound Mix w/ Fire

* *Setaria viridis* (L.) P. Beauv.: Invert

† *Sorghastrum nutans* (L.) Nash: w/o Organic

† *Spartina pectinata* Link: Intact, Invert, w/o Organic, Mix, Mound Mix

POLYGONACEAE

* *Rumex crispus* L.: Mound Mix

PORTULACACEAE

* *Portulaca oleracea* L.: Intact, Mound Mix, Mound Mix w/ Fire

SOLANACEAE

Solanum ptycanthum Dunal.: Intact, Invert, w/o Organic, Mix, Mound Mix, Intact w/ Fire, Invert w/ Fire, Mix w/ Fire

ULMACEAE

Ulmus rubra Muhl.: Mound Mix

URTICACEAE

Boehmeria cylindrical (L.) Sw: Mound Mix w/ Fire

Pilea pumila (L.) A. Gray: Mound Mix, Intact w/ Fire

Urtica dioica L.: Mound Mix

Appendix B Annotated Species List

List of species encountered in soil treatments during Chapter 2 field survey. Species binomial followed by survey season and mound number in which the species occurred: Pre-move – survey conducted in July 2014 before soil transfer; Post-move – survey conducted in August 2015 after soil transfer. Mound number is within parenthesis. Nomenclature follows ITIS (2016). An asterisk (*) indicates non-native species (USDA NRCS 2016). A dagger (†) indicates species that were included on the Little River Wetlands Project seed list.

ALPIACEAE

* *Daucus carota* L.: Post-move (1)

AMARANTHACEAE

* *Chenopodium album* L.: Pre-move (1,2); Post-move (1,2)

Chenopodium standleyanum Aellen: Pre-move (2)

ASTERACEAE

Ambrosia artemisiifolia L.: Pre-move (2); Post-move (1,2)

Ambrosia trifida L. Post-move (1)

* *Circium arcense* (L.) Scop.: Pre-move (1,2); Post-move (1,2)

Heliopsis helianthoides (L.) Sweet: Pre-move (1,2)

* *Lactuca serriola* L.: Pre-move (2)

Ratibida pinnata (Vent.) Barnhart: Post-move (1,2)

† *Solidago riddellii* L.: Pre-move (2)

Symphyotrichum sp. Nees: Pre-move (1,2)

Symphyotrichum dumosum (L.) G.L. Nesom: Post-move (1,2)

Symphyotrichum lanceolatum (Willd.) G.L. Nesom: Pre-move (1)

Symphyotrichum puniceum (L.) A. Löve & D. Löve: Pre-move (2)

* *Taraxacum officinale* F.H. Wigg.: Pre-move (1,2); Post-move (1)

† *Vernonia gigantea* (Walter) Trel.: Pre-move (2); Post-move (1,2)

BRASSICACEAE

* *Brassica rapa* L.: Post-move (1)

* *Erysimum cheiranthoides* L.: Pre-move (1,2)

CYPERACEAE

† *Carex hystericina* Muhl. Ex Willd: Post-move (1)

FABACEAE

† *Senna hebecarpa* (Fernald) H.S. Irwin & Barneby: Post-move (1)

LAMINACEAE

† *Mondarda fistulosa* L.: Pre-move (2); Post-move (1,2)

† *Pycnanthemum virginianum* (L.) Rob. & Fernald: Pre-move (2)

MALVACEAE

* *Abutilon theophrasti* Medik.: Pre-move (1,2)

PHYTOLACCACEAE

Phytolacca americana L.: Pre-move (1,2)

PLANTAGINACEAE

† *Penstemon digitalis* Nutt. Ex Sims: Post-move (1)

* *Plantago major* L.: Pre-move (1)

POACEAE

† *Andropogon gerardii* Vitman: Pre-move (1)

† *Panicum virgatum* L.: Pre-move (1)

* *Setaria faberi* R.A.W. Herrm.: Pre-move (1,2)

† *Spartina pectinate* Link: Pre-move (1); Post-move (1,2)

POLYGONACEAE

Persicaria pensylvanicum L.: Pre-move (1,2); Post-move (1,2)

* *Rumex crispus* L.: Pre-move (1,2)

PORTULACACEAE

* *Portulaca oleraceae* L.: Pre-move (2)

OXALIDACEAE

Oxalis stricta L.: Pre-move (1); Post-move (1)

ROSACEAE

Agrimonia sp. L.: Pre-move (1)

Fragaria virginiana Duchesne: Pre-move (2); Post-move (1)

Geum canadense Jacq.: Pre-move (1)

Rubus sp. L.: Pre-move (2)

URTICACEAE

Boehmeria cylindrical (L.) Sw.: Post-move (1,2)

Pilea pumila (L.) A. Gray: Pre-move (1,2)

Urtica dioica L.: Pre-move (2); Post-move (1)

VITACEAE

Vitus sp. L.: Post-move (1,2)